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AQUATIC RATIONAL THRESHOLD VALUE (RTV) CONCEPTS FOR ARMY ENVIRO--ETC(U)
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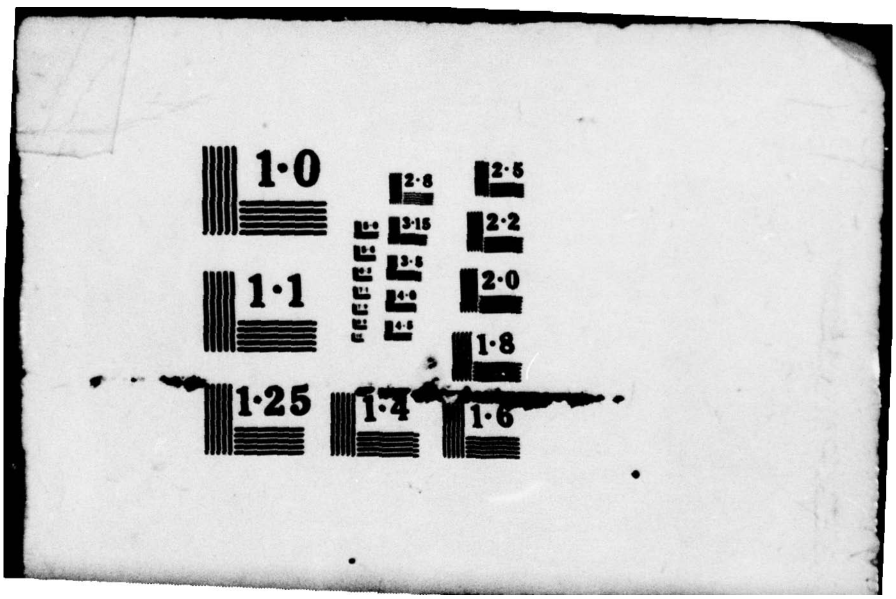
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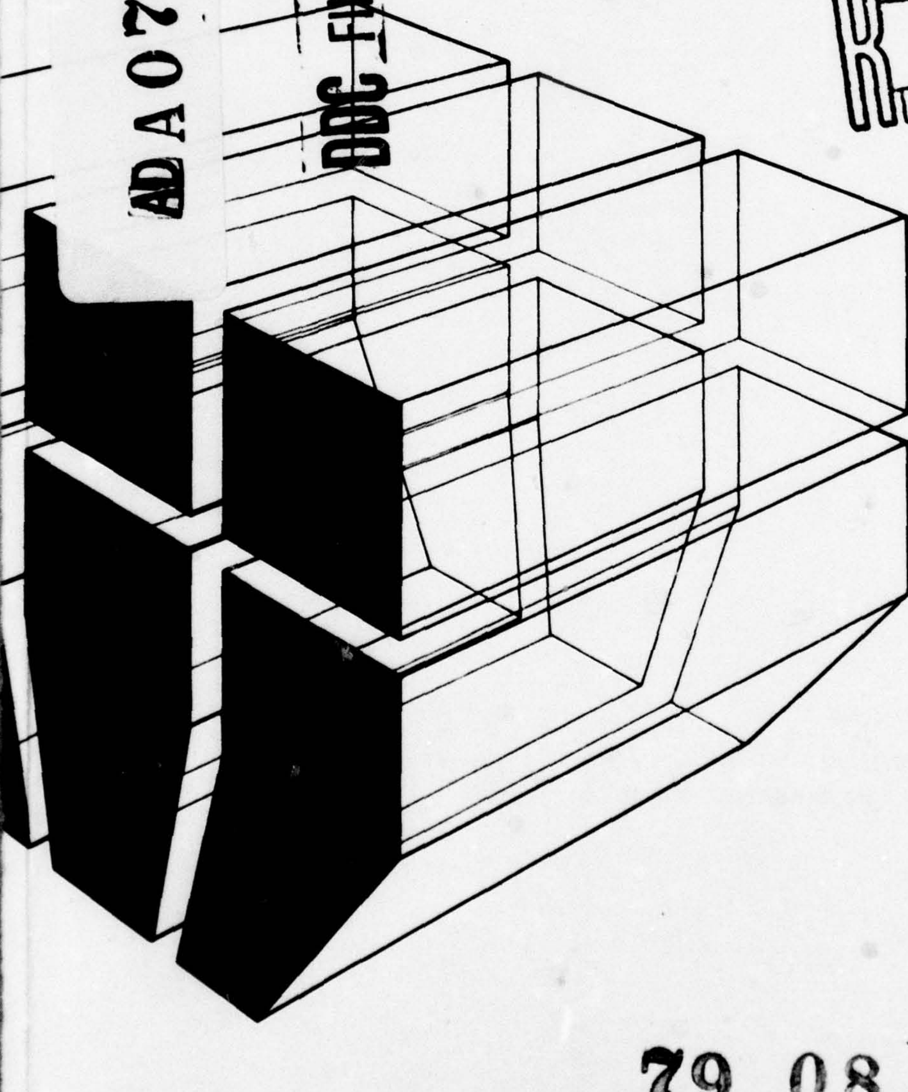
TECHNICAL REPORT N-74
July 1979

AQUATIC RATIONAL THRESHOLD VALUE (RTV) CONCEPTS FOR ARMY ENVIRONMENTAL IMPACT ASSESSMENT



by
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E. D. Smith

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This report presents the results of a study undertaken to develop practical techniques for evaluating the "significance" of predicted water quality impacts. Issues concerning the definition and use of the term "significant" are discussed at length. Existing aquatic ecosystem models are reviewed, potential criteria for measuring significance are examined, and a concept framework is presented. The result is a Rational Threshold Value (RTV) concept for measuring the significance of impacts on aquatic features. Initial development of an aquatic RTV involves determination

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of toxicity levels resulting from the introduction of pollutants into aquatic ecosystems, and expression of the effects of these toxicants on population levels of selected fish species.

FOREWORD

This study was performed by the Environmental Division (EN) of the U.S. Army Construction Engineering Research Laboratory (CERL) for the Directorate of Military Programs, Office of the Chief of Engineers (OCE) under Project 4A762720-A896, "Environmental Quality for Construction and Operation of Military Facilities"; Task A, "Environmental Impact Monitoring, Management, Assessment, and Planning"; Work Unit 006, "Analytical Model Systems for Prediction of Environmental Impacts." Mr. Paul Carmichael, DAEN-MPE-T, was the OCE Technical Monitor.

The work that led to the development of the information in Chapters 3, 4, and 5 was performed by Dr. E. E. Herricks and Mr. M. J. Sale (Department of Civil Engineering, University of Illinois, Urbana, IL) under contract No. DACA 88-77-M-0170 (Development of Rational Threshold Values for Aquatic Ecosystems).

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AQUATIC RATIONAL THRESHOLD VALUE (RTV) CONCEPTS FOR ARMY ENVIRONMENTAL IMPACT ASSESSMENT

1 INTRODUCTION

Background

Since the enactment of the National Environmental Policy Act (NEPA) in 1969, much progress has been made in the area of environmental impact assessment and planning.[1]* The early comprehensive interdisciplinary assessment methodologies, beginning with the work of Leopold,[2] have since been modified, culminating in very sophisticated computer-based analysis packages.[3,4] The literature provides several reviews of available methodologies and their applications.[5-7]

The initial assessment methodologies dealt with qualitative techniques. More recently, however, the President's Council on Environmental Quality (CEQ) has emphasized using an analytic rather than an encyclopedic approach to environmental impact analysis.[8] AR 200-1 establishes procedures for assessing the environmental impact of Department of the Army actions. CEQ guidelines are used as general guidance for preparation of Army EISs. Thus, more quantitative information is required.

The aquatic environment is an area often affected by new military projects or actions. Three principal components compose the aquatic environment: physical, chemical, and biological. An analytic approach can be used in all three component areas. An analytic approach requires models to generate quantitative information. It also requires a method to relate model output to impact significance.

An analytic approach requires measurable indicators of impact significance. To determine such significance, threshold values must be established. Therefore, it is necessary to develop concepts for using rational threshold values (RTVs) to measure the significance of impacts within the aquatic environment.

Objective

The objective of this study was to develop RTV concepts for establishing the significance of impacts on attributes of the aquatic ecosystem caused by Army military activities. These concepts will be used (1) to develop new approaches to the quantification and significance measurements of project environmental impacts, and (2) to establish both the basis for using RTVs and the framework for aquatic ecosystem RTV development.

Approach

Issues concerning the definition and use of the term "significant" were reviewed. Factors influencing the development of RTV systems for Army use were examined in terms of objective, operational and modeling constraints. Existing aquatic ecosystem models were reviewed, potential RTV criteria were examined, and a concept framework for aquatic RTV development was formulated.

Mode of Technology Transfer

These RTV concepts will be incorporated into analytical models for water quality which are now being developed. User manuals for water quality models to be issued in the DA Pamphlet 200 series will include appropriate instructions for use of these concepts. Water quality models and RTV will eventually become part of the Environmental Technical Information Systems.

2 "SIGNIFICANCE" IN ENVIRONMENTAL IMPACT ANALYSIS

Definitions

The term "significance" is used for different purposes, and there is no general consensus on its meaning with respect to environmental impacts. NEPA requires the preparation of Environmental Impact Statements (EISs) whenever Federal actions result in *significant* environmental impacts. The new CEQ regulations[9] require an environmental consequences section of an EIS to discuss the *significance* of a project's direct and indirect impacts. The regulations define the term "significantly," as outlined in Table 1. Although severity of impact is one criterion used to indicate significance, most other criteria are related to type of impact rather than to some quantitative measure. Factors to be considered include public health or safety, proximity to important land areas, environmental controversy, environmental uncertainty, precedence establishment, and cumulative effects.

*References are contained in the listing on pp 32 through 37.

Table 1

CEQ Definition of "Significantly"

Sec. 1508.24 *Significantly*

"Significantly" as used in NEPA requires considerations of both context and intensity:

(a) Context. This means that significance of an action must be analyzed in several contexts such as society as a whole (global, national), the affected region, the affected interests, and the locality. Significance varies with the setting of the proposed action. For instance, in the case of a site-specific action, significance would usually be a function of the effects in the locale rather than in the world as a whole.

(b) Intensity. This refers to the severity of impact. Responsible officials must bear in mind that more than one agency may make decisions about partial aspects of a major action. The following should be considered in evaluating intensity:

(1) Impacts that may be both beneficial and adverse. A significant effect may exist even if the Federal agency believes that on balance the effect will be beneficial.
(2) The degree to which the proposed action threatens public health or safety.
(3) Unique characteristics of the geographic area such as proximity to historic sites, park lands, prime farm lands, wetlands, wild and scenic rivers, or ecologically critical areas.
(4) The degree to which the effects on the quality of the human environment are likely to be highly controversial.
(5) The degree to which the possible effects on the human environment are highly uncertain or involve unique or unknown risks.
(6) Whether the action may establish a precedent for future actions with significant effects or represents a decision in principle about a future consideration.
(7) Whether the action is related to other actions with individually insignificant but cumulatively significant impacts. Significance exists if it is reasonable to anticipate a cumulatively significant impact on the environment. Significance cannot be avoided by terming an action temporary or by breaking it down into small component parts.
(8) Whether the action may have a significant adverse effect on an area or site listed in or eligible for listing in the National Register of Historic Places or may cause loss or destruction of significant scientific, cultural, or historical resources.
(9) Whether the action may have a significant adverse effect on the habitat of a species by the Endangered Species Act of 1973 determined to be critical.
(10) Whether the action threatens a violation of Federal, State, or local law or requirements imposed for the protection of the environment.

Since neither NEPA nor its implementing guidance provides a practical, working-level definition of significance, other means must be used, as shown by the simple example in Figure 1. Assume that a population undergoes a natural (unaltered by human activity) fluctuation with time. Assume further that human activity will occur at time T_0 and will cause an estimated initial decrease (Delta P) in population level. Delta P is the magnitude of the population decrease. Is the impact significant? Figure 2 shows several possible ramifications of an initial population decrease. Line OL is the estimated population change with time under natural conditions. Line OSL represents a situation of eventual recovery to historical average levels. Line OSL′ is recovery to a lower average level, and OSL″ represents eventual loss of the species. The actual complexity of impact analysis is only partially shown in Figure 2. However, the figure provides several possible criteria which could be used to measure significance.

One possible criterion is historical levels. This assumes that beyond the historical low, an impact becomes significant. Another criterion—irreversibility—is favored by many scientists, but has disadvantages. Figure 2 illustrates two types of irreversibility. Line OSL″ represents species extinction. Criteria for significance could be some value approaching PM, the population level below which the species cannot be maintained, but what value should be used? Should half the distance between PA and PM be the point at which an impact becomes significant, or should it be 95 percent? Another type of irreversibility is shown by Line OSL′, where the species is maintained at a new level, PN. The effect may be irreversible, but, again, at what new level does the impact become significant?

Line OSL shows recovery after some time, TR. Several seasons of low-level populations of an important, hunted species could generate considerable controversy. Would this not be a significant impact?

Additional factors contributing to the complexity of the problem but not reflected in Figure 2 are discussed in the following paragraphs.

1. **"Real World" Complexity.** Figure 2 represents population change for a single species. In the "real world," the impact situation is more closely represented by an N-dimensional space in terms of N state variables. At any time, the state of the environment can be defined by a vector in N-dimensional space. At some future time, the effects of an action could result in a new environmental state, a gain represented by a vector in N-dimensional space. Therefore, using a single species as an indicator of impact significance is a considerable simplification.

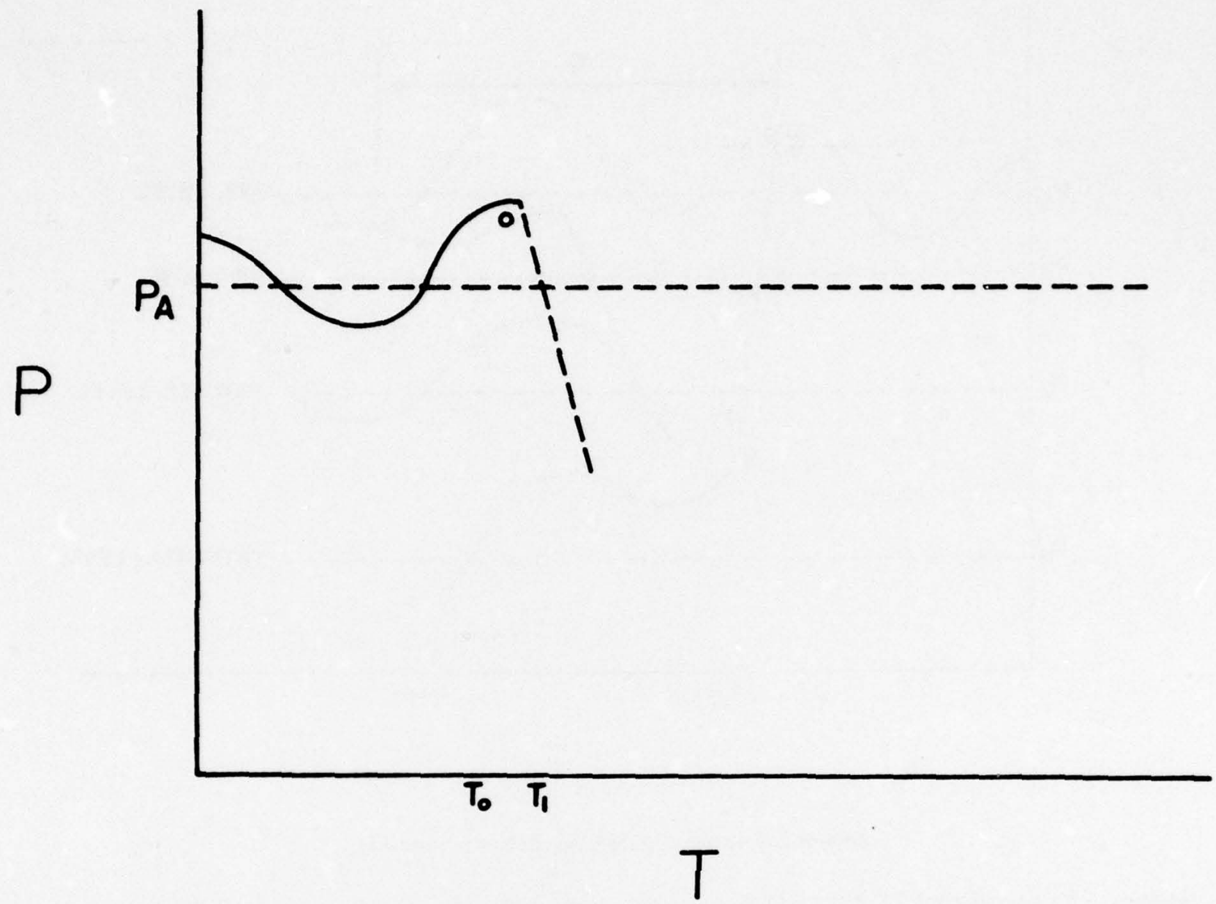

Figure 1. Population fluctuations resulting from some activity.

2. **Environmental Continuity.** The use of some value as the level at which an impact becomes significant introduces a binary variable (yes or no) into a process that varies continuously.

3. **Cumulative Effects.** Impacts that seem insignificant individually may be significant cumulatively or may represent one effect in a chain of impacts which cumulatively are significant or will eventually become significant.

4. **Total Picture.** The use of an indicator (or even several indicators) to determine significant impact levels may fail to account for the potential trade-offs between impacts and benefits. Although there may be significant adverse impacts on aquatic biota, other benefits (e.g., economic) may outweigh these effects. For example, all reservoir projects change aquatic ecosystems; however, a different perspective of a project's significance may be gained if a stream ecosystem is replaced with a reservoir ecosystem rather than just lost. It is necessary to differentiate between the significance of a single impact and the total impact of an action.

5. **Human Perspective.** Human perspective implies the significance of a project's impact on human welfare.

6. **Conservation Vs. Development.** There are differences of philosophy between resource conservationists and those interested in using these resources. For example, the loss of trees for lumber may be an adverse impact in the opinion of a conservationist but a benefit in the opinion of a developer.

7. **Spatial Context.** What may be a significant impact at a regional level often becomes insignificant when considered in national context. Events which are regionally insignificant could be of great local

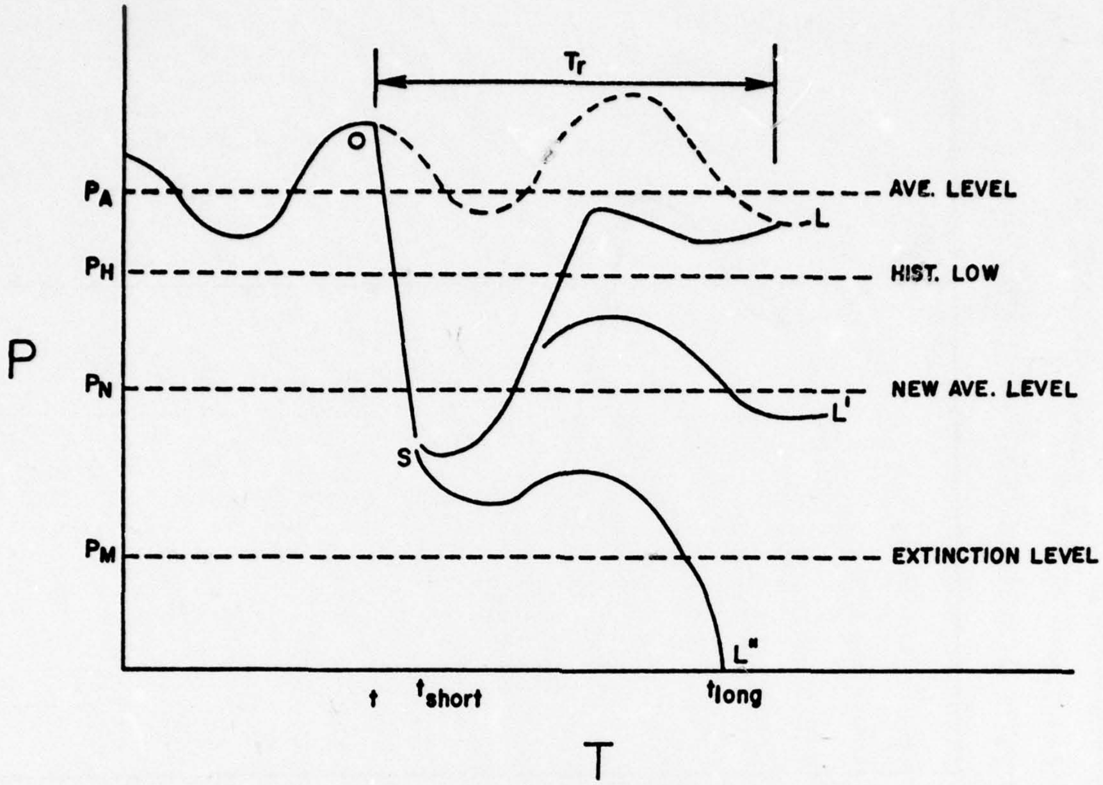

Figure 2. *Impact scenarios on species population.*

interest. The spatial context of an impact must be considered when judging its significance.

8. **Uncertainty.** Note in Figure 2 that all levels past Point 0 are estimates and that all the points before 0 require historical data which may be lacking. When selecting the level of impact significance, the user must consider that there is a great deal of uncertainty with impact estimation. Extreme natural environmental fluctuations can increase or decrease the effects of impacts caused by human activity. This introduces additional uncertainty into impact and significance prediction.

9. **Point of Irreversibility.** The point at which impacts become irreversible is very difficult to determine. The point at which proximity to irreversibility becomes significant is also unclear.

Although the previous discussion has neither defined the term "significance" nor identified criteria for measuring it, it does indicate the difficulty encountered with using scientific terminology or criteria to define "significance." Since environmental impact analysis is a decision-making process, a practical working-level definition of significance might best be developed from the process itself.

The Role of Significance in the Environmental Decision-Making (EDM) Process

Significance as it applies to environmental impacts first enters the EDM process when the environmental impact assessment is concluded. At this time, the EDM must either conclude that there is no significant impact or begin preparing an EIS. If an EIS is required, a scoping meeting must be held. This meeting determines the scope of issues to be addressed and identifies the significant issues. Participants are Federal, state, and local agency representatives, proponents of the action, and other interested persons. The participants determine the scope and the significant issues to be analyzed in depth in the EIS. Insignificant issues are eliminated from detailed study.

Thus, in the DM process, there are two objectives for determining the significance of impacts: (1)

whether an EIS should be produced, and (2) what scope of issues should be covered in the EIS. From this standpoint, significance can be associated with the interest and concern of the EDM and other interested parties.

The decision-maker is first interested in significance as an indicator of the need for an EIS, and next with selecting information to be contained in the EIS. These two needs are related; i.e., if environmental impact analysis reveals information of sufficient interest, then an EIS should be prepared. Therefore, the significance of direct and indirect impacts should be considered and a definition of significance developed that fits the decision-maker's needs. Consider the following definition: "A significant impact is that level of effect that generates such interest and concern on the part of interested parties that the decision-maker requires that the ramifications of the impact should be studied in detail and documented in the EIS." Thus, the decision-maker establishes the levels of effect considered to be significant; these levels, or measures of significance, are identified as RTVs. The fact that a particular level of impact does not reach the RTV does not preclude the probability that other scoping meeting participants want the impact to be addressed in the EIS.

3 RATIONAL THRESHOLD VALUES

RTV Conceptualization

RTVs measure the significance of environmental impacts. Examination of the etymology of "rational threshold value," specifically the meanings of rational, threshold, and value in the context of impacts on aquatic systems, provides additional insight into defining significance of impact.

There are two important aspects to defining "rational" when dealing with environmental assessment. The pertinent definitions taken from Webster are: "implies the ability to reason logically, as by drawing conclusions from inferences, and often connotes the absence of emotionalism," and "relating to or resulting from the application of arithmetic operations." The Rational Method for calculating stormwater runoff is a good example of using empirical equations in a logical analysis for design purposes. Unfortunately, when considering environmental matters, especially those which are difficult to quantify (e.g., aesthetics, integrity, etc.), the precise definition of "rational" must be modified. Major difficulties arise when there is no quantitative basis on which to base logically reasoned conclusions. Therefore, in an environmental context, "rational" must include matters which are well thought out, but which may contain a nonquantitative base, often one that is associated with "irrational" political or social factors.

For use in this RTV analysis of aquatic ecosystems, the question of rationality has been approached by attempting to quantify or at least uniformly apply subjective analysis to determine what is rational. In this regard, environmental concern which includes subjective or emotional judgments is important. For example, the loss of one species from an ecosystem may have little impact on the community's overall structure or function. The rational approach would require accepting that loss, and recognizing that overall community function could be maintained. However, if the lost species is designated as rare or endangered—a classification which by legislative fiat requires action for preservation—the approach has been to accept this type of external constraint and incorporate it in the analysis, whether it is rational or not.

Threshold, when used in the RTV concept, includes a variety of definitions. A threshold is defined as "*the beginning point of something . . . a stimulus* just strong enough to produce a response." In aquatic ecosystems, a threshold may be more than the first indication of a stress; it may be a point which, when crossed, will be impossible or very difficult to return to, i.e., irreversible impacts. In aquatic systems, both the possible cause-effect relationships which would bring the community to a threshold, and the mechanisms which would restore it to some level of structure or function typical of pre-impact conditions must be considered.

When considering stress effects in aquatic ecosystems, the natural variability in physical and chemical conditions and the lack of detailed knowledge about the system's aquatic biota often preclude a detailed definition of the threshold. If the stress is short-term, time-related changes in ecosystems are usually insufficient to cross a threshold for all components; thus, although damage may occur at the species or population level, ecosystem structural changes may be minor, and function can be maintained.

Defining "threshold" for aquatic ecosystems must be approached at various levels of biological or eco-

logical complexity. The first level would integrate an organism-specific response, basing threshold values on toxicity testing or evaluation. The second approach would integrate the organism-specific response, but extrapolate it to a population-level effect. Further extrapolation would integrate community response. The final approach is based on ecosystem-level integration. It has been argued that response curves at the ecosystem level are linear and therefore exhibit little or no threshold response. This integration requires knowledge not only of the ecosystem, but also of its interactive components, especially on a time-related base. In analyzing the recovery of aquatic ecosystems from stress, Cairns[10] has proposed a useful synthesis of ecosystem interactions. Cairns has defined and summarized several ecosystem relationships and assessed the system's vulnerability in terms of inertial, elasticity, and resiliency relationships. Ecosystem inertia includes the system's ability to resist displacement of structure or function, and elasticity implies its ability to recover from damage. Resiliency is related to the number of times the ecosystem can be stressed and still return to nearly normal structure or function. If these concepts of stress response relationships are used in an RTV analysis, defining "threshold" becomes quite complex; the definition will also be highly subjective because of a scarcity of data.

The term "value" places major constraints on RTV analysis and may have major responsibility for altering the perspective of the analysis procedure. "Value" implies some form of quantification, and it is extremely difficult to quantify certain components of aquatic ecosystems.

Constraints on RTV Development

Examining the constraints under which Army RTVs must operate can provide insight to the structure of an RTV system and the framework within which it must operate. Three types of constraints must be examined: objective, operational, and model. Objective constraints include such concerns as (1) the type of impacts for which RTVs should be developed, (2) what indicators of significance should be used, and (3) where the RTVs should be applied in the chain of interrelated effects that may result from an Army action. Operational constraints deal with such concerns as (1) user characteristics, (2) when and how RTVs might be used in the EIA process, (3) how RTV-related data should be developed and maintained, and (4) how the use of RTVs fits into the Environmental Technical Information System. Model constraints are also involved, since models are required to obtain the input data for use in RTV analysis.

Objective Constraints

Figure 3 illustrates the major water quality problems at a typical Army installation. Both point and area sources of water pollution are present. Within the aquatic ecosystem, there are several levels in the chain of interaction at which RTVs could be applied. Figure 4 illustrates a typical impact process. Tracked vehicle training (1) results in pollutant emissions which are transported to streams (2). Water quality is degraded (3) throughout the stream system (4) and affects aquatic biota (5). Should only direct impacts be subjected to RTV analysis, or should some other point in the chain be chosen? Water quality is a possible subject for RTV analysis. Water quality is dynamic in both time and space, but legal standards, such as NPDES permit stipulations, can be used as RTV criteria. Pollution levels reaching the stream could become criteria; however, because of modification effects of chemical parameters within the stream, it may be difficult to determine the ultimate effects. One advantage of using pollutant input as a criterion is that it identifies the effects at or close to the source. Initial development of RTV should be limited to point sources, since this simplifies pollution analysis. Consideration must be given to dispersion aspects in any water quality modeling effort.

Figure 5 illustrates the complicated interactions of water quality attributes and aquatic biota. Additional complexity is introduced by including interactions among water quality attributes (Figure 6). Problems could arise because certain water quality attributes are site-specific. Which attributes are more important? It may be best to begin with the end points of the chain; i.e., either introduction of pollutants or effects on aquatic biota.

For example, there are several trophic levels to consider for aquatic biota. RTVs could be developed for different levels within the food chain. Higher levels are more visible and generally better understood; however, lower levels would be effective as RTV criteria since serious effects can be identified here before they affect higher levels in the food chain. However, at these low levels, little is known about the effects of pollution and other activities. Assuming that the higher food levels are chosen (e.g., fish species), the user encounters another problem in choosing an indicator species. Again, the more popular game species are more visible; how-

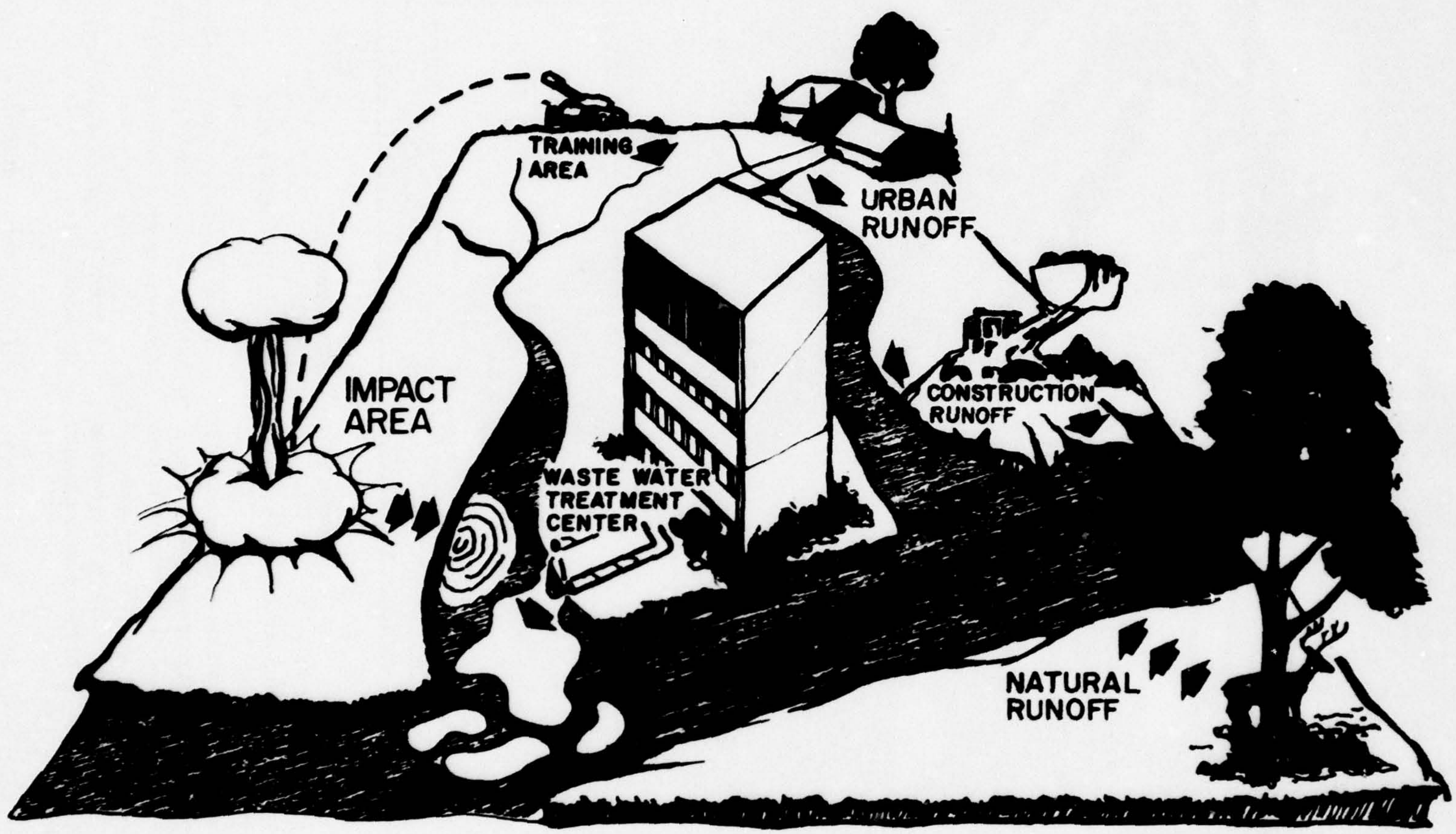

Figure 3. Typical water quality problems.

Figure 4. Example impact process.

ever, other species may be more important to community maintenance.

The type of RTV criteria used to indicate significance should match the target ecosystem parameter chosen as an indicator. Most of the factors involved in choosing a certain type of RTV were discussed earlier in this chapter. An RTV can be expressed as the function of one or more factors and take the form of Eq 1.

$$RTV = f(H, HS, L, C, IR, RV, CU, B, P, I, SC, \ldots) \quad \text{[Eq 1]}$$

where H is historical trends.

HS is related to public health and safety. The exact value of this factor is a function of data accuracy, completeness, and length of record.

L refers to legal standards and is, essentially, a given criterion.

C represents environmental controversy which is itself a function of many factors, such as interest in the impacted parameters, interest in benefits from

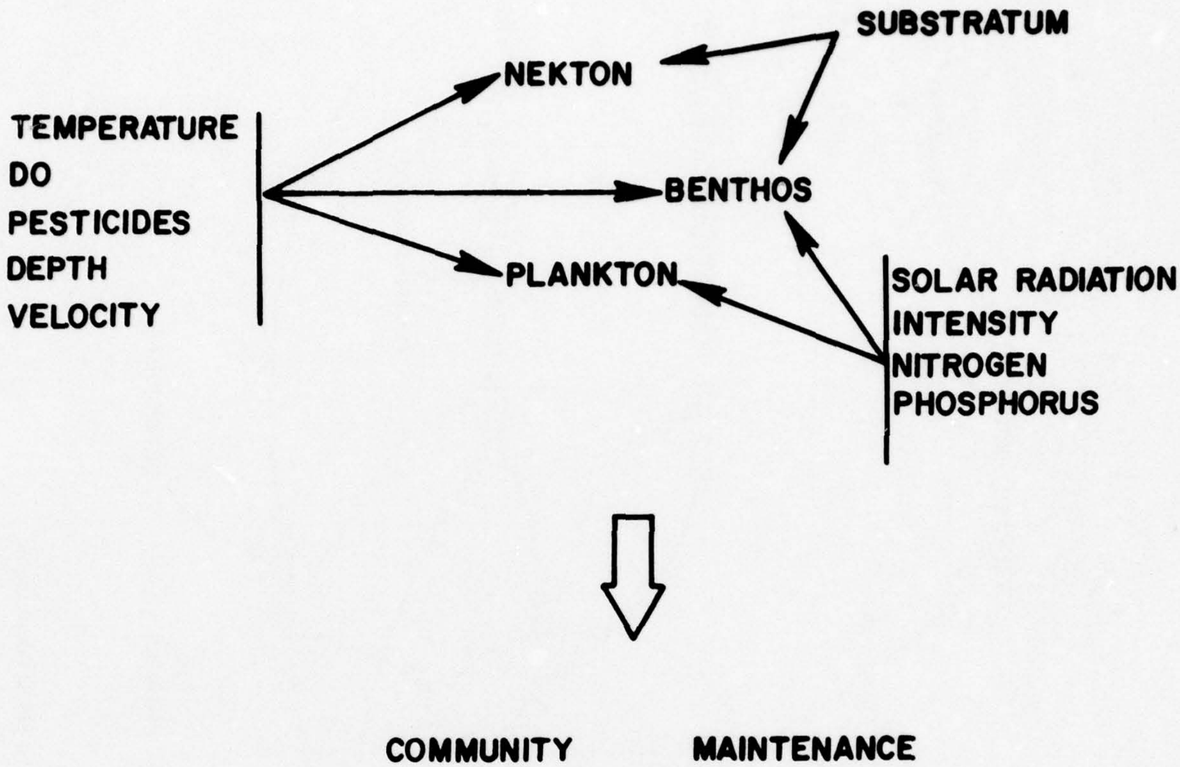

Figure 5. Water quality/aquatic biota relationships.

the action, information dissemination, interested organizations, etc.

IR represents irreversibility, which is a function of different ways to express irreversibility, difficulty of determining when irreversible levels are reached, and disagreement about how close an impact can come to the irreversible limit before the impact is significant.

RV represents the relative value of an impacted attribute when compared to other similar attributes present, the degree to which the attribute is affected by the action, and offsetting benefits of the action.

CU represents cumulative effects.

B represents the possibility for future actions which may also have effects.

P is related to the proximity to unique resources.

I represents the interrelationship of an impacted attribute with other ecosystem attributes.

SC indicates that significance must be established with reference to spatial context. The function is left open because many other factors could be involved. Figure 7 shows the interrelationships of various criteria.

Operational Constraints

Operational constraints involve factors pertaining to using RTVs, such as user capabilities. For example, RTVs must be oriented for use by individuals having little or no practical knowledge of aquatic ecosystems. Simplicity and minimum input data are also requirements, and most important, the RTVs must provide information useful to the decision-making process.

Another important factor is the method by which the RTV will interface with other impact analysis procedures. Figure 8 shows the Environmental Technical Information Systems (ETIS), the cornerstone for Army systematic procedures for impact analysis. RTVs should be designed to be compatible with this system. RTVs are also to be used with a system of

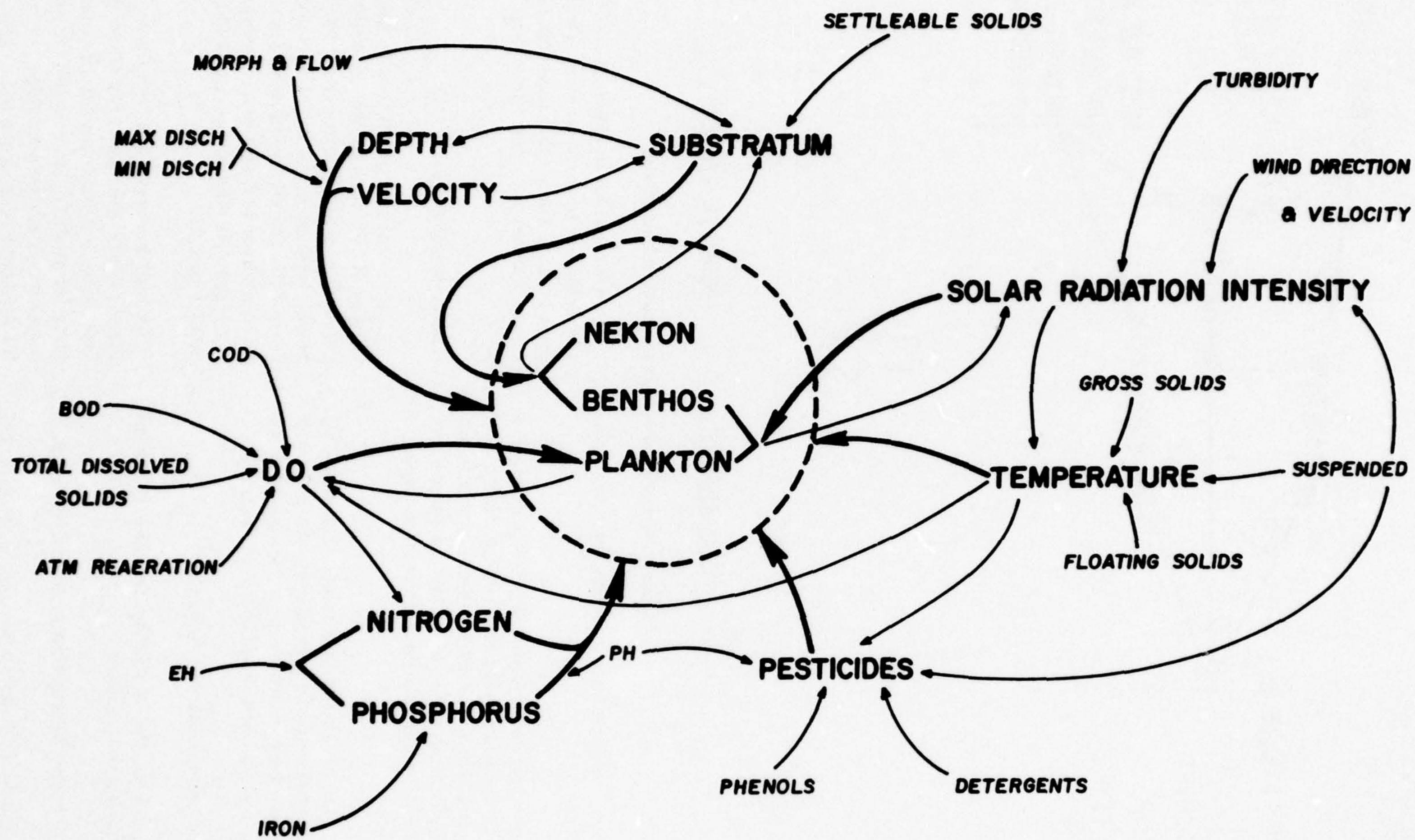

Figure 6. Complexity of aquatic interactions.

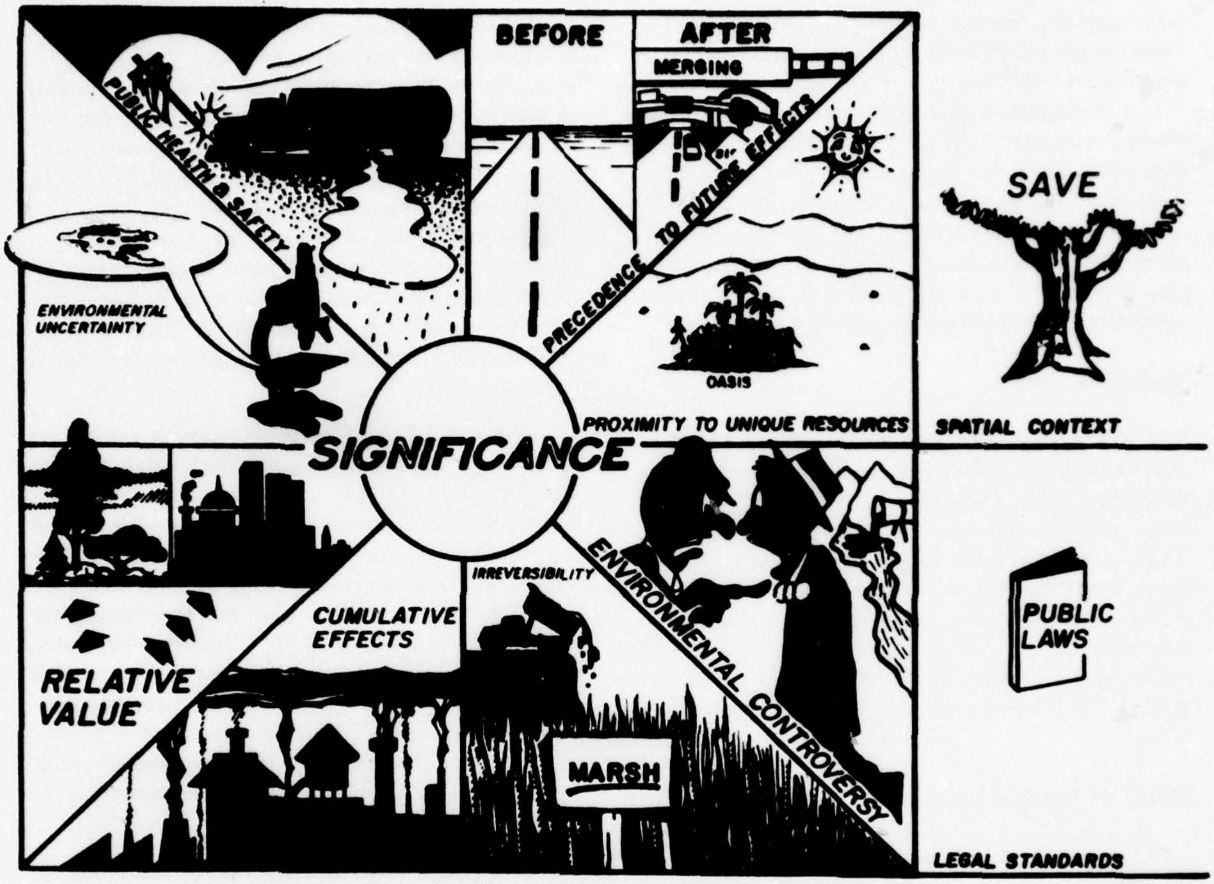

Figure 7. Criteria interrelationships.

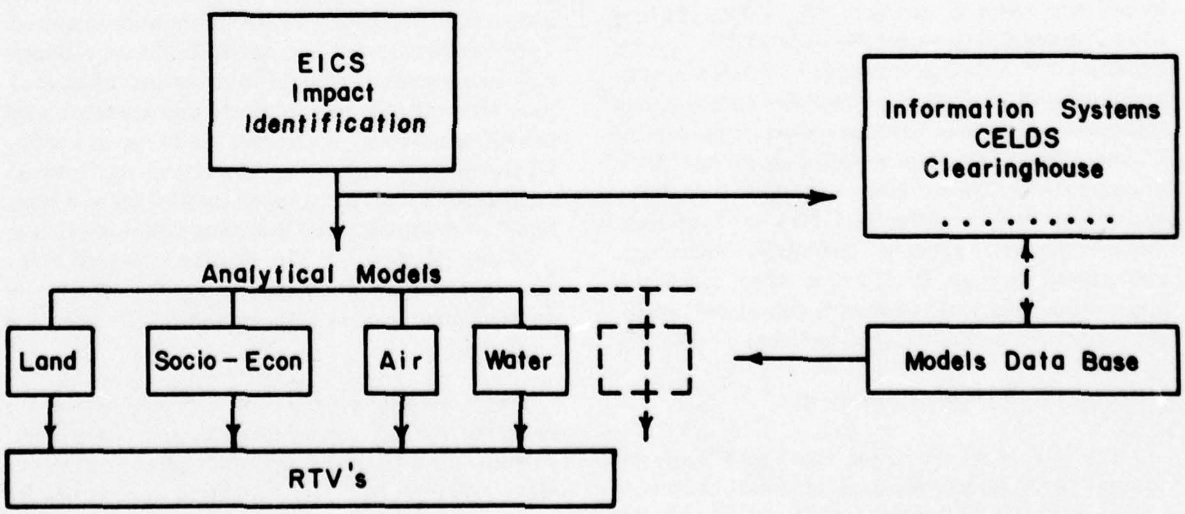

Figure 8. ETIS structure.

water quality models now being developed. The selection of models to be used and model complexity are of great importance.

The development of an RTV system must also include consideration for data maintenance; potential expansion using new stressors, activities, or indicators; the frequency of data update and the difficulty of data acquisition; storage and retrieval of data; comprehensiveness of use among varied situations and ecosystems; and model degree of resolution, calibration, fine tuning, accuracy, and precision.

Model Constraints

It is the goal of analytical impact modeling to represent the quantitative relationships between project activities and preliminary impacts, and between primary impacts and higher-order impacts. RTVs can be applied at any point of the impact chain. The RTV does not operate independently, but rather is used to assess a project impact in conjunction with output from some predictive model. The following sections discuss the status of analytical models which may be suitable for use with the RTV concept.

Status of Aquatic Ecosystem Models

In little more than a decade, the science of ecosystem modeling has grown to become a major branch of ecology. The requirements for impact assessment by NEPA and the research efforts of the International Biome Program have provided major impetus for this growth. Modeling of aquatic environments has received much of this attention, and several general reviews of the state of the art are available.[11-18] Although there are still differences in approach and controversy in aquatic modeling, one statement which would receive universal agreement is that aquatic ecosystem modeling is an interdisciplinary activity. The dynamics of aquatic ecosystems encompass the scientific disciplines of hydrology, thermodynamics, aqueous chemistry, toxicology, and aquatic ecology. In addition, when man's use and misuse of aquatic systems is considered, socioeconomics and politics must be included.

Modeling the Physical Environment

There are three principal components of the aquatic environment: physical, chemical, and biological. Of these, the mathematical models for the physical component are the most advanced. The engineering discipline of hydrology is a well-developed design science capable of providing detailed descriptions of surface water flows.[19] Grimsrud,[13] et al., have given an excellent review of available simulation models. Existing models can easily predict mean stream discharge based on watershed area and local hydrologic conditions (rainfall patterns, runoff relationships, etc.) and estimate depth and velocity parameters of stream flow as a function of instantaneous discharge. These models have received widespread usage.[20,21] The main problem with hydrologic modeling of large watersheds is that extensive calibration (fine tuning of the models) is often necessary.

Sedimentation engineering is closely related to hydraulic modeling. There are mathematical relationships for evaluating erosion rates, sediment transport, equilibrium-suspended solids loads, and depositional/scour zones and rates. Although these models are somewhat less reliable than well-calibrated hydraulic models, they are satisfactory for planning alternative actions.[22] Some of the more important sediment models are the Universal Soil Loss Equation and its modifications,[23] and the concept of Unit Stream Power.[24] Karr and Schlosser have reviewed the interactions between these models and the biological components of the aquatic ecosystem.[25]

Water temperature and available insolation at various depths can be estimated by calculating relatively simple energy balances on a body of water. The increasing concern over thermal discharges in aquatic environments has led to the development of many reliable heat models.[26-29] These models vary in complexity, depending on the information required. They have been used successfully to design diffusers for heated water discharges to rivers and lakes, evaluate thermal effluents in rivers and marshes, and model vertical and horizontal gradients in cooling lakes. Available subsurface insolation can be simulated easily by using extinction coefficients as a function of water quality and incoming solar radiation as a driving function.[30,31] The shading effects of riparian vegetation on small streams can be modeled in an analogous fashion with an additional extinction coefficient.

These models provide an excellent basis for modeling the hydrologic, thermal, and photic components of an aquatic ecosystem's physical environment. Complex and fairly exacting models can be constructed, and simplified modeling methods are available which have both a reasonable degree of reliability and less extensive data requirements.

Modeling the Chemical Environment

The basis of all conservative chemical water quality modeling is the mass balance equation.[16,18,12] Given mass inputs of a conservative compound of interest from such sources as surface runoff, point discharges, upstream advection, and/or bottom sediment sources, bulk concentrations can be calculated temporarily or spatially when flow velocity and dispersion coefficients are known. The accuracy of the calculations depends primarily on the simplifying assumptions which have been made. Unfortunately, many of the chemical species of interest are not conservative in nature. Biological conversions, atmospheric exchanges, sediment exchange, precipitation and dissolution, and radioactive decay all provide sources and sinks for particular chemical compounds within the water column. In almost all cases, there are no acceptable methods for modeling the dynamics of nonconservative elements.

The relationship between dissolved oxygen (DO) and biochemical oxygen demand (BOD) has received more attention than any other phenomenon in the aquatic environment, beginning with Streeter and Phelps.[32] There are many models for simulating this system, which reflects not only linkages to hydraulic and temperature components of the physical environment, but also linkages to the nitrogen cycle.[33] Table 2 provides an assessment of the capabilities of existing modeling applications. In addition, biological rate constants for the aquatic system can be estimated fairly accurately. This type of impact assessment is now being used in Illinois to evaluate variances for sewage discharge regulations. DO/BOD modeling is an example of what can be accomplished with ecosystem simulations.

If it is accepted that the mass balance equation is adequate for modeling conservative compounds in the chemical environment, the main problem to be faced is how to handle the sources and sinks of nonconservative compounds within the water column. For the inorganic compounds involved in chemical reactions, such as shifts in the carbonate system, or other precipitation reactions, equilibrium modeling can be used to indicate at least the trend of chemical dynamics.[31,34,35] Although equilibrium models cannot give information on the rate of reactions, they can establish the boundary conditions toward which the chemical environment will proceed. Models for predicting mineral equilibria have been developed and are being used at Stanford University. These could be adaptable to impact analysis. Many examples of this type of analysis of the pH-alkalinity-carbonate system are available. Toxic chemicals (as well as minerals) can be modeled in this way. Once the absolute concentration of toxicant is calculated, its impact can be analyzed via the toxicity unit concept.[36-39]

Table 2

Capability of Existing Modeling Applications

Water Quality Characteristics	**Level of Current Analytical Approaches**	
	Level I*	**Level II****
Dissolved Gases - O_2, N_2, CO_2		x[a]
Temperature		x
Sediment		
Suspended	x[c]	
Bedload	x[c]	
Total Dissolved Solids		x[b]
Nutrients	x[d]	
Detritus	x[e]	
Toxic Materials	x	
Bacteria		
Pathogens	x	
Decomposers	x	
Algae		
Planktonic	x	
Sessile	x	
Macrophytes	not available	
Macroinvertebrates	x[f]	

*Level I - low to moderate accuracy, less precise

**Level II - highly accurate, precise

[a]With the exception of benthic O_2 production and demand

[b]With the exception of chemical phenomena such as $CaCO_3$ solution and precipitation

[c]With the exception of methods for channel change effects (blank sloughing, aggradation migration)

[d]Techniques at Level II are available in many situations

[e]Limitations with measurement and characterization

[f]Available only with extensive and careful data acquisition

Modeling the Biological Environment

The least well-developed component of aquatic ecosystem simulations is modeling of the biological environment. While biological modeling is advancing rapidly, acceptable population models exist only at the extremes of the trophic organization. The main source of simulation routines for aquatic populations comes from subroutines in large-scale ecosystem models which have been developed recently.[18,31,40-43] Although many of these ecosystem models have not been constructed specifically for use in impact analysis, their analytical formulations are applicable in a general format.

Microbial growth dynamics are usually represented in the form of Monod or Michaelis-Menten kinetics. Specific growth rates and half-saturation coefficients have been documented and are available for several types of algae and bacteria. Microbial respiration rates can be linked to actual or simulated water temperatures through the Q_{10} relationship.[31] Generally, these models are more suitable for projecting future trends than for simulating exact population levels. This evaluation applies to periphyton as well as phytoplankton.

Fisheries models are at approximately the same stage of development as the phytoplankton models.[44-46] Fisheries models have already been used as impact analysis tools to evaluate power plant operations;[47-49] most of these are developed from analytical models (as opposed to harvest models) similar to the work of Kitchell, et al.[50] As for the phytoplankton models, the literature reviews many of the species-specific model parameters. Most models are set up to evaluate mortality imposed on a population through entrainment, impingement, or heat shock. However, the potential to link population projections to other habitat changes such as increased chemical toxicity or physical habitat changes is good. The use of toxicity indices[41] in age-specific population models as described by Jensen[51] may have direct application to RTV analysis.

Ecosystem components in the trophic level between algae and fish are less well modeled. This category includes macrophytes and benthic populations for which combined biomass estimations are the best projections possible. On a community level, there are no methods available to project diversity or other parameters of community structure. The use of the Saprobic system of species diversity has been applied to evaluate existing environmental damage, but these concepts have not been included in ecosystem simulations. These ideas have not been applied because systems analysis of aquatic systems requires large-scale combination of species dynamics, and because the cause-effect relationship between diversity and stability is not known.

Existing Aquatic Ecosystem Models

Several comprehensive ecosystem models incorporate all of the previously mentioned components required for analytical impact analysis. One of the most flexible and extensive[12] has been developed by the U.S. Army Corps of Engineers.[50] This model is applicable to both river and reservoir systems and can simulate one-dimensional temperature stratification, BOD, several trophic levels of fish, benthos, zooplankton, algae, detritus, organic sediments, phosphorus, total dissolved carbon, NH_3, NO_3, NO_2, O_2, coliforms, alkalinity, TDS, light penetration, and pH. This is an example of the highest available level of aquatic ecosystem models.

4 EVALUATION OF POTENTIAL CRITERIA FOR RTVs

Ecosystem models may be quite complex. The major determinants of model complexity are the required output and the availability of data (required for both model calibration and operation). If the full range of ecological interactions at all levels (or even several) of physical, chemical, and biological organization are to be modeled, the difficulty associated with model use for environmental assessment activities is obvious. Nevertheless, it is possible to use established ecosystem principles to expand state-of-the-art assessment methodologies. The pragmatic view taken in this report recognizes the limitations placed on model use by data and manpower constraints, and by insufficient knowledge of environmental variables; however, models have been used by carefully constructing a set of simplifying assumptions that supplement existing data bases with predictions of ecosystem dynamics.

The criteria discussed in this chapter consolidate environmental setting and activity information into quantitative indices which can be used to assess projected impacts in aquatic ecosystems. This chapter identifies analytical approaches from existing ecosystem models which can be used to simplify and augment EIA procedures. Table 3 summarizes possible RTVs, their inputs, outputs, and the impact problems they address.

Water Quality Indices

Generalized water quality indices (WQI) have been proposed to describe overall environmental conditions in aquatic ecosystems.[52,53] These indices serve as a composite informational parameter which indicates a water body's degree of pollution. A WQI is essentially a weighted function of several different water quality parameters. These functions can be either additive or multiplicative.

$$WQI_a = w_{ai} P_i \qquad \text{[Eq 2]}$$

Table 3

Summary of Possible RTVs

RTV	Impacts Addressed	Inputs	Outputs	References
Water Quality Index (WQI)	Overall water quality	DO, fecal coliforms, pH, $NO_{3\text{-}N}$, $PO_{4\text{-}P}$, BOD_5, temperature, total solids, turbidity	Relative condition of overall water quality	1,2
BOD/DO	Organic, point source pollution—mainly in rivers and streams (may include NOD and SOD)	Stream channel morphometry, temperature, BOD, biological rate constants, point source discharges, stream discharge	Oxygen deficits, instream concentration of NH_3, NO_3, BOD, DO, etc.	3
Saprobic Index (SI)	Organic, point source pollution—mainly rivers and streams	BOD_5	Saprobian classification of biological communities	4,5
Trophic State Index (TSI)	Eutrophication—mainly in lakes and reservoirs	Transparency (Secchi Disk), [CHla], [total phosphorus]	Indication of lake trophic condition	6
Nutrient Loading Models	Eutrophication—mainly in lakes and reservoirs	Basin morphometry, phosphorus inflow and outflow, stream flow, land use	Projected lake trophic condition	7,8
Autotrophic Index (AI)	Eutrophication—both lentic and lotic environments	Nutrient concentration, carbonate system, light, temperature	Relative dominance of autotrophic component of microbial community	9
Relative Algal Growth Potential (RAGP)	Eutrophication—both lentic and lotic environments	Nutrient concentration, carbonate system, light, temperature	% of maximum growth rate for components of algal community, limiting environmental parameters	
Toxicity Unit (TU)	Environmental toxicity	Indicator species specific LC50, modifying factors (i.e., hardness, etc.)	Overall acute toxicity of environment	10,11
Population Growth Index (PGI)	Impacts on reproduction and survival	Age specific fecundity and survival functions	Net population reproductive rate per generation (idealized)	12
Population Simulations	Cumulative and long-term effects on higher levels of trophic structure	Population parameters such as fecundity and survivorship and age structure of initial standing crop	Projected levels, stability, recovery rates from short-term impacts	13,14

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$$WQI_m = \Pi w_{mi} P_i \qquad \text{[Eq 3]}$$

where P_i = i^{th} descriptive water quality parameter considered
W_{ai}, w_{mi} = weighting constants
a = additive
m = multiplicative
i = 1 through the total number of parameters.

Water quality parameters suggested for consideration in these indices include dissolved oxygen, fecal coliforms, pH, NO_3-N, PO_4-P, BOD_5, temperature, total solids, and turbidity.

Threshold values for WQI would be difficult to determine because of the variability allowed in individual parameters while maintaining the same index value. Although the data requirements for identifying water quality condition could be met by using existing monitoring programs or STORET* data bases, the analytical data required to include necessary impact variables in a WQI would be beyond the capability of most assessment activities. WQIs may be useful if integrated with other indices (e.g., toxicity units).

Dissolved Oxygen/Biochemical Oxygen Demand

Probably the oldest method for assessing the impact of point source discharges on aquatic environments is the use of dissolved oxygen models, as used in the work of Streeter and Phelps.[32] Although these models are usually used to simulate downstream concentrations of BOD and DO, the mathematical equations on which they are based can be manipulated to provide information even more useful for RTVs. A preliminary step to calculating a total sag curve should be the analysis of t_c, the time of flow to the point of minimum oxygen concentration, and D_c, the maximum oxygen deficit expected from a given discharge.

$$t_c = \frac{1}{k_2 - k_1} \ln \left\{ \frac{k_2}{k_1} \left[1 - \frac{k_2 - k_1 D_o}{k_1 L_o} \right] \right\} \qquad \text{[Eq 4]}$$

where k_1 = BOD rate constant
k_2 = reaeration constant
D_o = initial dissolved oxygen deficit
L_o = total BOD loading rate

$$D_c = \frac{L_o k_1}{k_2} \exp(-k_o t_c).$$

This approach will delineate the maximum impact and its spatial location and indicate whether further analysis is necessary.

The RTV for D_c would be a function of local water quality standards and water temperature. If D_c violated this RTV, a full simulation could be made of the DO sag curve for further analysis. Dissolved oxygen models can be modified to account for transverse diffusion[54] or for a more complete consideration of ecosystem functions such as photosynthesis, benthic oxygen demand, and nitrogenous oxygen demand.[37] Choice of models depends largely on data availability.

Saprobic Index

The Saprobian system and its modifications have been used to classify the trophic condition of aquatic ecosystems since the beginning of this century.[55-60] This system basically uses the principle of indicator species as a descriptive measure of the organic pollution's impact on aquatic communities. The Saprobic Index, S, is usually designated by a number from 1 to 4 or 1 to 8 and is associated with degrees of trophic conditions ranging from pure water (katharobity) to lifeless liquors (ultrasaprobity). The application of saprobic indices has generally been used as a classification scheme in Europe, but has not been widely used in the United States.

*USEPA Water Quality Storage and Retrieval System

Calculating a Saprobic Index directly from taxonomic identifications is beyond the skill and capability of typical environmental assessment activities. However, numerous authors have shown the relationship between Saprobic Index and instream BOD concentrations.[61-63] Table 4 gives relative values of S and BOD_5 as taken from Sladecek.[59,60] In addition, the following analytical relationship can be used to derive S from BOD_5:

$$S = \frac{k\,S_o(L - L_o)}{1 + k(L - L_o)} \qquad \text{[Eq 5]}$$

where L = 5-day BOD
S = Saprobic Index

Sladecek and Tucek[63] derived this predictive equation and estimated the values of the constants under two conditions.

	k	S_o	L_o
$0 < \le BOD_5 < \le 50$ mg/ℓ	0.218	4.93	0.44
$BOD_5 > 50$ mg/ℓ	0.0021	9.0	−420

Although this equation is strictly empirical, it has the potential to provide additional information for an EIA, especially when combined with a BOD/DO model. BOD_5 is a commonly measured water quality parameter, and can also be related to other water quality parameters such as COD. Water quality models such as the Streeter-Phelps model can predict downstream concentrations of BOD_5. The calculation of S can provide a relative estimate of impacts on the receiving stream's biological community including downstream successional patterns and the spatial extent of the impact of organic materials discharged from point sources.

The application of an RTV to the BOD/S model shown above is academic in many instances where there are already instream water quality standards for BOD concentrations. Tables such as Table 4 give enough information for setting approximate threshold values. Where BOD originates from multiple sources, both natural and man-made, this model can give information concerning cumulative environmental impacts.

Trophic State Index

Carlson[64] has recently developed several simplified methods for calculating indices of the trophic conditions of lakes. The trophic state index (TSI) can be independently calculated from three different water quality parameters: Secchi disk transparency, chlorophyll a concentrations, or total phosphorus concentrations.

$$TSI_{SD} = 10\left(6 - \frac{\ln SD}{\ln 2}\right) \qquad \text{[Eq 6]}$$

$$TSI_{Chl} = 10\left(6 - \frac{2.04 - 0.68 \ln Chl}{\ln 2}\right) \qquad \text{[Eq 7]}$$

$$TSI_{TP} = 10\left(6 - \frac{\ln 48/TP}{\ln 2}\right) \qquad \text{[Eq 8]}$$

where SD = Secchi disk (m)
Chl = chlorophyll a concentration (mg/m^3)
TP = total phosphorus concentration (mg/m^3).

The use of TSI values can provide estimates of the relative eutrophy of lakes and reservoirs. TSI values range from 0 to 100.

TSI	Trophic Condition
0 to 40	Oligotrophic
40 to 50	Mesotrophic
50 to 100	Eutrophic

These values of TSI are consistent with the ranges of Secchi disk readings and chlorophyll a and total phosphorus concentrations used by the National Eutrophication Survey (NES) to classify surface waters.[65] The analytical relationships used by Carlson to develop the TSI formulas shown above were derived from data taken from a limited number of lakes, so caution should be exercised when using them. Additional data from NES could easily be used to further verify these relationships.

The TSI approach to classifying surface waters is relatively simple to understand and use. To use a

Table 4

Values of S and BOD_5 (= L) for Upper Limits of Individual Saprobic Degrees

Degree	S	L	Note
Katharobity	−0.5	0.0	Purest water
Zenosaprobity	0.5	1.0	Very clean
Oligosaprobity	1.5	2.5	Clean
Beta-mesosaprobity	2.5	5.0	Mild pollution
Alpha-mesosaprobity	3.5	10.0	Pollution
Polysaprobity	4.5	50.0	Heavy pollution
Isosaprobity	5.5	400.0	Sewage
Metasaprobity	6.5	700.0	Septic
Hypersaprobity	7.5	2,000.0	Putrefaction
Ultrasaprobity	8.5	120,000.0	Lifeless liquors

TSI as a predictive tool for RTV analysis, it is necessary to incorporate variables which can be associated with project activities having environmental impacts. In the case of TSI_{TP}—the trophic state index calculated from total phosphorus concentrations—this can be accomplished by considering watershed land uses and point source discharges. Use of methods for including impact variables within TSI_{CHl} or TSI_{SD} will be much more difficult.

Nutrient Loading Indices

Many models for analyzing nutrient inputs to lakes have been developed during the past few years.[66-71] The use of these models to assess phosphorus loading rates has been reviewed by Gakstatter et al.,[65] and Tapp.[72] The utility of these models is very high in the RTV context. Tapp concluded that the simplified loading models provided essentially the same impact information as large-scale simulation models. The work of Dillon[67] is an example of such a model's potential for use in an RTV.

Dillon's phosphorus loading model, which was presented in graphical form, consisted of a plot of the quantity $L(1 - R)/p$ versus $\bar{z}$, the mean basin depth. In this equation, L is the total annual phosphorus loading rate ($gm/m^2/year$), R is the phosphorus retention coefficient (that fraction of input not lost in output), and p is the mean hydraulic flushing time (exchanges/year). Regions of this graph corresponding to oligotrophic, mesotrophic, and eutrophic conditions were identified and verified with NES and other real data. The definition of these trophic conditions can also be put into the following analytical form:

$$P = \frac{L(1 - R)}{\bar{z}p} \quad \text{[Eq 9]}$$

where P = estimated steady-state phosphorus concentration.

P	Trophic Condition
<10	Oligotrophic
10 to 20	Mesotrophic
>20	Eutrophic

The information required for calculating Dillon's index can often be supplied, using USGS hydrologic data and water quality monitoring data from state or Federal environmental protection agencies. Thresholds of impact are defined from lake trophic status. The impacts of both point and nonpoint source (due to land use alterations) loadings of phosphorus can be assessed as they affect downstream water bodies. A series of loading indices could be developed from information in references listed in footnotes 66-71; the index used would be dependent on information availability. This approach is similar to that proposed by Tapp.[72]

Autotrophic Index

Weber has proposed the use of an Autotrophic Index (AI)—the ratio of organic matter to chlorophyll a—to monitor impacts on aquatic ecosystems.[73] These parameters can be measured in terms of mg/m^3 for phytoplankton communities or mg/m^2 for periphyton communities, resulting in a dimensionless index value. AIs are responsive to changes in the microbial communities downstream from municipal wastewater discharges in the Ohio River.[69] AI values are also a good indicator of a wide range of impacts which affect the relative autotrophy/heterotrophy of aquatic systems.

The simulation of AI, as opposed to in situ measurement, can provide the basis of an RTV for assessing impacts on aquatic ecosystems' trophic dynamics. Essential data would come from population models of the major primary procedures (algae) and decomposers (bacteria). Minimal inputs would be the major limiting factors at algal growth, including nutrients (C, N, and P), insolation and temperature, and some determinant of bacterial population growth, such as BOD or total organic carbon. AI can then be formulated in one of the following ways.

$$AI = \frac{\text{dry weight of organic material}}{\text{chlorophyll a}}$$

$$= \frac{\text{algae + bacteria + detritus}}{\text{k} \cdot \text{algae}}$$

$$= \frac{1}{k} + \frac{\text{bacteria}}{\text{k} \cdot \text{algae}} + \frac{\text{detritus}}{\text{k} \cdot \text{algae}} \quad \text{[Eq 10]}$$

These parameters can represent weights per unit area or volume basis. The analytical models required to simulate population levels of algae and bacteria and detritus concentrations were reviewed previously (pp 19 and 20). Generally, these models are not adequate to predict exact population levels; however, the structure of this proposed index is in itself an important tool for evaluating combinations of environmental variables and the ways in which they may affect biological communities. The fact that this

index incorporates ecological information favors an AI over an index such as WQI in the decision-making processes.

Data for constructing, testing, and evaluating an AI model are readily available from STORET files and from sampling stations of the National Water Pollution Surveillance System. Information for deriving RTVs from AI results is available from the same source and from Weber.[73]

Relative Algal Growth Index

A more simplistic approach than the AI model for assessing environmental impacts on primary producer populations is the simulation of algal growth potential as a function of pre- and post-project conditions. In its most simplified form, this model could be structured on Monod kinetics.

$$\hat{\mu} = \mu_m Q_{10}^{\left(\frac{T-20}{10}\right)} \left(\frac{L}{K_L + L}\right)\left(\frac{C}{K_C + C}\right)\left(\frac{N}{K_N + N}\right)\left(\frac{P}{K_p + P}\right) \quad \text{[Eq 11]}$$

where $\hat{\mu}$ = specific algal growth rate
μ_m = maximum specific algal growth rate
T = water temperature
L = insolation
C = concentration of total available inorganic carbon
N = concentration of total nitrogen
P = concentration of total phosphorus
Q_{10} = a temperature coefficient which relates reaction rates at different temperatures

$$Q_{10} = \left(\frac{K_{T1}}{K_{T2}}\right)^{10} \Big/ (T_1 - T_2)$$

T_1 and T_2 are two different temperatures.
Typical values of Q_{10} are between 1.02 and 1.06.

K_L, K_C, K_N, K_p = ½ saturation constants; K_i is the concentration of i which produces ½ the maximum growth rate.

Without knowing μ_m, the ratio of $\hat{\mu}/\mu_m$ can be calculated and used to represent the Relative Algal Growth Index (RAGI). Although more sophisticated formulations for the temperature and nutrient relationships are available, the analytical relationship developed represents the growth regulation of algal populations; this information can be used to assess limiting factors to algal growth. When limiting factors are identified, it is possible to develop RTV values. This approach can be used to evaluate shifts in the dominance of different algae groups; and in turn, these data can be used as input to the water quality models discussed previously.

Toxicity Unit Index

The concept of toxicity units is a useful technique for integrating biological response to toxic compounds with environmental modifying factors such as ambient dissolved oxygen, temperature, pH, etc.[44,74] The toxicity unit, TU, is a measure of the acute environmental toxicity resulting for one or more toxicants present in the aquatic environment. TUs are built up from combinations of various compounds that are specific for each target species. The concentration of each toxic compound present is weighted by its 96-hr-LC50 to calculate toxic units. No synergistic or antagonistic effects are taken into account, mainly because there is a lack of information concerning cumulative effects of multiple toxicants.

$$TU = \sum_{i=1}^{NT} \left(\frac{c_i}{LC50_i}\right) \quad \text{[Eq 12]}$$

where c_i = measured or predicted concentration of i^{th} toxic compound present (mg/ℓ)
$LC50_i$ = 96 hr-LC50 for i^{th} compound as modified by environmental conditions (mg/ℓ)
NT = number of toxicants present.

An important consideration in using TUs is that both the c_i and the $LC50_i$ can be functions of environmental setting parameters. For example, ionization can change the effective concentration of a toxicant such as ammonia, or ambient water hardness can affect the realized LC50 of many toxic metals. When the analytical relationships of these effects are available, they can easily be incorporated into a TU model. The values used for $LC50_i$ are also species-specific. In this sense, the TU index is a function of the target species present in the geographical region of each project. A series of TUs can be calculated for each target species designated as part of a project's environmental setting.

TU models have the potential to provide useful information about the relative degree of environmental toxicity; TU output indicates the proportion of total toxicity for which each toxicant is responsible, and gives a fair representation of regional specificity. Further development of this index may provide a valuable tool for condensing environmental information into input appropriate for a decision-making mode. Problems inherent in this model include: (1) toxicity information (i.e., LC50 values) is available for a limited number of toxicants under varying ambient water quality conditions, (2) relatively few indicator species have been tested, and (3) the cumulative effects of multiple toxicants are rarely additive. Threshold values for mixtures of toxicants are also difficult to quantify, although investigators have provided some guidance for British waters.[41] Despite these problems, TU models may be an important part of an RTV methodology. In the future, as research provides additional toxicity information, the use of TU models may find wider application. The linkage of conservative and nonconservative element water quality models with TU models to provide predictive analysis of environmental toxicity would be especially valuable.

Population Growth Index (PGI)

The potential growth rate for natural populations of species exhibiting distinct developmental stages (i.e., age classes) has been represented as[28,29]

$$R_o = \sum_{x=1}^{NA} l_x m_x \qquad \text{[Eq 13]}$$

where R_o = rate of population increase per generation

l_x = survivorship of individuals from age class 0 to x

m_x = mean number of offspring produced by individuals of age x − 1 to x

NA = number of age classes in population.

When additional sources of mortality (other than natural) are included in this equation, this relationship can become the basis of an RTV for assessing impacts on higher organisms. The obvious RTV is to maintain $R_o \geq 1$ in order to assure conditions suitable for nonnegative population changes. This is a new model which has not been proposed for assessing environmental impacts. However, Eq 13 has been used extensively to manage populations that are subject to harvesting.[75] In those cases, the requirement to maintain $R_o \geq 1$ is referred to as a sustained yield constraint. It is relatively simple to progress from considering environmental impacts which affect population survivorship and reproductive rates as a form of harvesting to using such established methods of harvest management for assessment purposes.

Analytical equations which can describe the cause-effect relationships between environmental perturbations and changes in survivorship and fecundity must still be developed. Available data from toxicity testing/dose response curves are one source of this type of information. A potential problem is the inability to estimate availability of l_x and m_x parameter estimation for natural populations. Nevertheless, development of a PGI model may provide a desirable compromise between TU models, which contain less information, and complete population simulations. Future work on PGI models should consider habitat alterations, stochastic representation of population parameters, and modeling of invertebrate populations.

Population Levels

For many higher organisms such as fish and some invertebrates, simplified population models can be used to simulate the cumulative effects of environmental impacts in aquatic ecosystems. This approach has been proposed previously[76-78] and has been implemented to investigate entrainment and impingement impacts from power plants.[79,56] Generalized computer models which can incorporate the large number of impact mechanisms necessary for a flexible EIA tool have not yet been developed. This problem can usually be traced to the fact that models are too often developed for site-specific problems.

Generalized population simulation models which can handle a variety of impact mechanisms can be constructed using the Leslie matrix and its modifications.[80-82] The advantages of these matrix models are that they are computationally straightforward, easily adaptable to computerization, and applicable to established analysis techniques.[83,84] In addition, matrix models can be constructed to simulate not only density-independent, deterministic birth and death processes, but also stochastic processes,[85] migration phenomena,[86] optimization analysis,[87] and density population control. These models have not been sufficiently developed for use in EIA analysis.

The most important role of population criteria in relation to other RTV criteria is that a flexible population model (e.g., for a target fishery) can integrate cumulative impacts and show how they ultimately affect higher trophic levels in terms of long- and short-term productivity. Questions of population stability and elasticity can be addressed either by conducting time simulations of standing crops[55] or by examining the population projection matrix structure.[78] Reversible and irreversible impacts can be differentiated by population model response after removing project activity effects after a given period of time.

One of the most useful population simulation models is the Leslie matrix.[80] The basic structure of the Leslie matrix model is

$$N_{t+1} = A \cdot N_t \qquad \text{[Eq 14]}$$

where N_t and N_{t+1} = column vectors of dimension *n* which represent the age-specific population structure (number of individuals in each class) for the t and t+1 time periods

A = nxn square matrix

$$A = \begin{bmatrix} f_0 & f_1 & \ldots & f_{n-1} & f_n \\ S_0 & 0 & \ldots & 0 & 0 \\ 0 & S_1 & \ldots & 0 & 0 \\ 0 & 0 & & S_{n-1} & 0 \end{bmatrix}$$

in which S_x = age-specific survivorship rate for individuals of age class x (proportion surviving from age x to x+1)

f_x = age-specific fecundity for individuals of age x to x+1 (mean number of offspring produced per individual in age class x)

This model often represents the female portion of a population, but appropriate corrections for sex ratio can be made. It is then trivial to show

$$N_t = A^t N_0 \qquad \text{[Eq 15]}$$

This population model represents exponential growth with no density-dependent or density-independent growth regulation. However, it is possible to revise the basic project matrix elements to make the model more responsive to EIA needs. Each f_x and S_x term can be modified by coefficients representing both the project activities' impacts and their consequences.

$$f'_x = DDF_x \cdot DIF_x \cdot f_x$$

and

$$S'_x = DDS_x \cdot DIS_x \cdot S_x \qquad \text{[Eq 16]}$$

where DDF_x = density-dependent control coefficient of fecundity of age class x

DIF_x = density-independent control coefficient of fecundity of age class x

DDS_x = density-dependent control coefficient of survivorship rate of age class x

DIS_x = density-independent control coefficient of survivorship rate of age class x

f_x, S_x = national fecundity and survivorship rates of population in specified environmental setting.

f'_x and S'_x are then used for the elements in the projection matrix. DDS_x and DDF_x terms are functions of population densities. DIS_x and DIF_x are functions of impact variables such as discharges of toxic compounds, physical and chemical habitat parameters, and other direct or indirect harvests of individuals. It can be seen that if impact variables such as these are incorporated into f'_x and S'_x, all of the analyses described above can be carried out at any point along a simulation time line. Further research remains to be done concerning the possibilities of using readily available information such as LC50 and Maximum Acceptable Toxic Concentration (MATC) values, behavioral avoidance reactions, and electivity data as the basis for calculating DIS_x and DIF_x.

Community Indices

Indices of community structure (e.g., numbers and density of species) have largely been summarized by "diversity indices." These indices are sensitive to pollution and thus applicable to RTV classification, although their predictive capability is severely limited. The literature in this field largely revolves around the mathematical approaches for calculating biological index values for whole communities. Wilhm[88] used a modification of information theory analysis to calculate the diversity index (d) for a community and then to assign typical ranges of d for conditions ranging from clean to

highly polluted. Cook[39] who reviewed the sensitivity of these various indices in polluted and unpolluted situations, found that a modification of the Chandler score[40] was the most sensitive index. Kaesler and Herricks[41] also discuss the validity of the two commonly used information theory analyses and conclude that the modification of the Shannon-Weiner index as proposed by Wilhm and by Olive and Smith[42] lacks sensitivity to low-density samples. This brief review of biological community indices indicates that few are reliable in all situations; thus, to provide a meaningful and ordered system having general application to RTV analyses would require a combination of several approaches.

The information content in the typical community diversity index may be quite large, depending on the investigator's expertise. However, several adptations of community indices do not require trained analysis personnel, and these may be useful in RTV analysis.

5 AQUATIC RTV CONCEPTS

The discussion and criteria presented in Chapters 2, 3, and 4 provide a basis for developing RTV concepts; however, the following important factors should also guide RTV development.

1. An impact becomes significant when it reaches a level that generates interest and concern. Effective RTVs can only be established by a general consensus among interested parties. This is difficult and may take some time. Meanwhile, the decision-maker must be responsible for determining when an impact becomes significant.

2. The complex interactions among attributes of aquatic ecosystems make it extremely difficult to establish RTVs within the chain of effects. Therefore, initial aquatic RTVs should be developed to measure the significance of cumulative impacts on the higher, more visible trophic levels.

3. Since spatial and temporal aspects of area source pollution are more complex than aspects of point source pollution, initial RTVs should address point source pollution. This also implies that initially, impacts from pollution emissions will be considered before direct physical impacts.

4. By definition, RTVs (1) are established not arbitrarily but with careful consideration and informed judgment (rational); (2) represent a "yes" or "no" condition (threshold) pertaining to the significance of impact; and (3) are quantitative (values).

5. RTVs should be used with output from analytical models.

Two types of RTVs satisfy the requirements of the factors listed above: water quality standards and population levels.

Water Quality Standards as RTVs

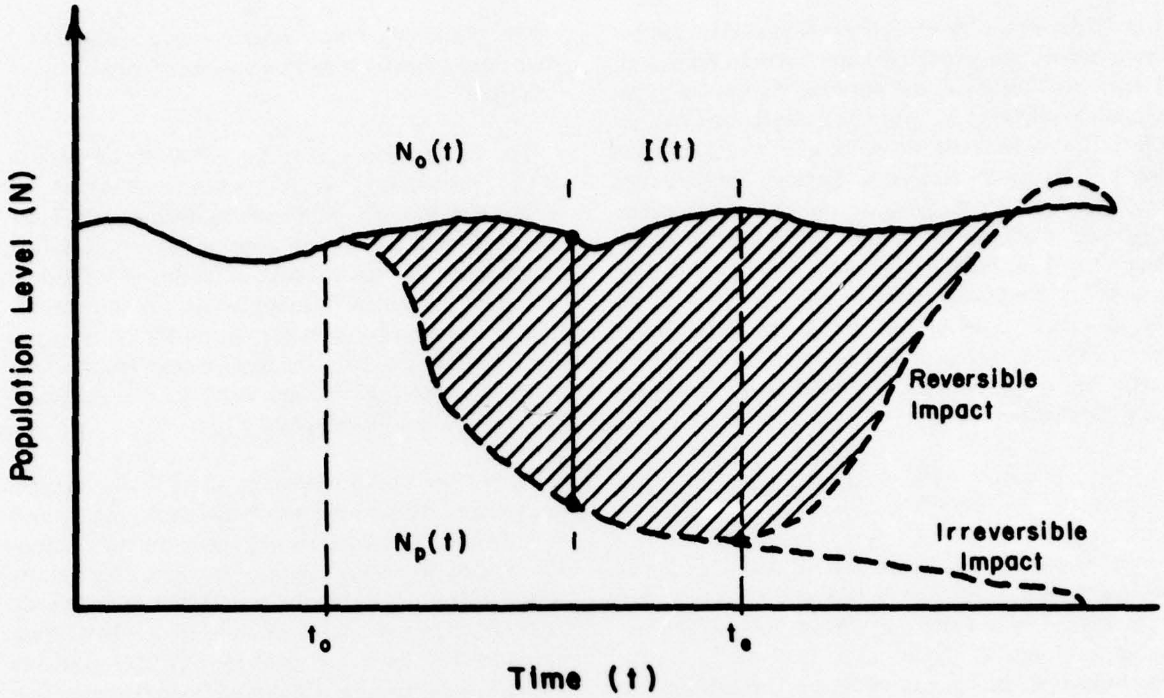
Water quality standards are practical and necessary criteria for measuring impact significance. RTVs can easily be associated with water quality models; their use requires developing a data base of water quality standards for streams at and around Army military installations.

Population Levels as RTVs

Population levels satisfy many RTV requirements and are well suited to initial RTV development. Fish are excellent indicators, since they are of the highest trophic level, and toxicity is an excellent indicator of initial source of stress. Water quality and toxicity models and data to drive them are available.

Using population models for EIA predictions requires a clear understanding of model output. Output of population models is an index of impact rather than an actual deterministic prediction of future standing crops. The goal of these models is to represent the general directions and relative magnitudes of environmental impacts. Because of the data and fiscal resource limitations of most impact assessment activities, population models are not exact predictors. Similarly, environmental impacts affect aquatic ecosystems at the community level; it is therefore an oversimplification ecologically to represent single population of organisms independent of their competitors, predators, and prey. However, despite these limitations, population level simulations can provide valuable impact assessment information. Output from population simulations becomes a prototype for evaluating impact magnitude weighing the costs and benefits of alternative project designs, and minimizing ultimate overall impacts.

Several important quantitative definitions of environmental impacts can be developed from population model output. Figure 9 illustrates the utility of population-level impact prediction. Impact magnitude, I(t), is measured by comparison to a "no-



t_o = time at which project activities are initiated

t_e = time at which all project activities have ended

$N_o(t)$ = baseline population simulated with environmental setting and no project activity specifications

$N_p(t)$ = impacted population simulated with environmental setting plus project activity specifications

$I(t)$ = impact at time t = $N_p(t) - N_o(t)$

Figure 9. Example of population simulation for impact assessment.

project" baseline simulation. Values of I(t) greater than zero can represent a net beneficial impact (assuming that the population simulated was that of a desirable species). Population-level impacts can be standardized by calculating the relative impact, RI(t), as the ratio of I(t) to the baseline without the project, $N_o(t)$. Threshold values can then be placed on RI(t) to designate "significant" impacts (i.e., $RI_{RTV}(t) = -0.9$, meaning that the original population has been depressed to 90 percent or less of its original level at time (t). Population stability can be defined using this terminology as

$$S = 1/\int_{t_o}^{\infty} (RI(t))^2 dt \qquad \text{[Eq 17]}$$

where S = population stability.

If multiple populations at various trophic levels can be simulated, community stability can also be defined as in Eq 18.

$$S_c = 1/\frac{1}{M} \sum_{i=1}^{M} w_p(t) \int_{t_o}^{\infty} w_t(t)\,[RF_i(t)]^2\,dt \qquad \text{[Eq 18]}$$

where M = number of populations simulated
$w_p(t)$ = weighting function for population
RI_i = relative impact on the population
$w_t(t)$ = weighting function for time period
S_c = community stability
dt = change in time.

Definitions for reversibility of impacts can be developed as indicated in Figure 9. After removal of impacts at time t_r, if I(t) approaches zero as time approaches infinity, an impact can be termed reversible. If the limit of I(t) does not approach zero, the impact is termed irreversible. Further analysis can be carried out by recognizing the class of impacts which are managerially reversible[93] and examining the effect of restocking or reclamation plans on the recovery of simulated populations. The concept of elasticity, E, of a population, the ability to return to normal after being displaced by environmental impacts, can be approximated by the rate of recovery of the simulated population.

$$E(t) = \frac{d\,I(t)}{dt} \quad \text{for } t > t_r \qquad \text{[Eq 19]}$$

With the use of the Leslie matrix model, population parameters such as stability and elasticity can be measured in another way. When the population projection matrix is primitive,[94] a positive real eigenvalue can be calculated which corresponds to finite population growth rate, R. The remaining eigenvalues of the project matrix are negative or complex and can be used to describe the stability of the population.[95]

$$S' = |\lambda_{max}| - \frac{1}{M+1} \sum_{i=0}^{M} |\lambda_i| \qquad \text{[Eq 20]}$$

where $|\lambda_i|$ = $\sqrt{x_i^2 + y_i^2}$ = modulus of i^{th} eigenvalue
x_i = real part of i^{th} eigenvalue
y_i = complex part of i^{th} eigenvalue
$M + 1$ = number of eigenvalues = number of age classes
S' = stability of simulated population.

It should now be clear that although population-level simulations may be limited in their ability to represent real-world ecosystem dynamics, they do have important advantages in terms of quantitative assessment techniques. Rigorous definitions of impact types and mechanisms developed in the context of simulation model outputs cannot be derived easily in any other manner. Impacts in this case resulted from point source discharges of BOD, and the analytical model used the BOD/DO equation. Threshold values were defined in terms of oxygen deficits; optimization of the mitigation loop involved iterations to reach a point where those thresholds were not exceeded (DO less than water quality standards) while at the same time optimizing some management objective (cost minimization or equity maximization).

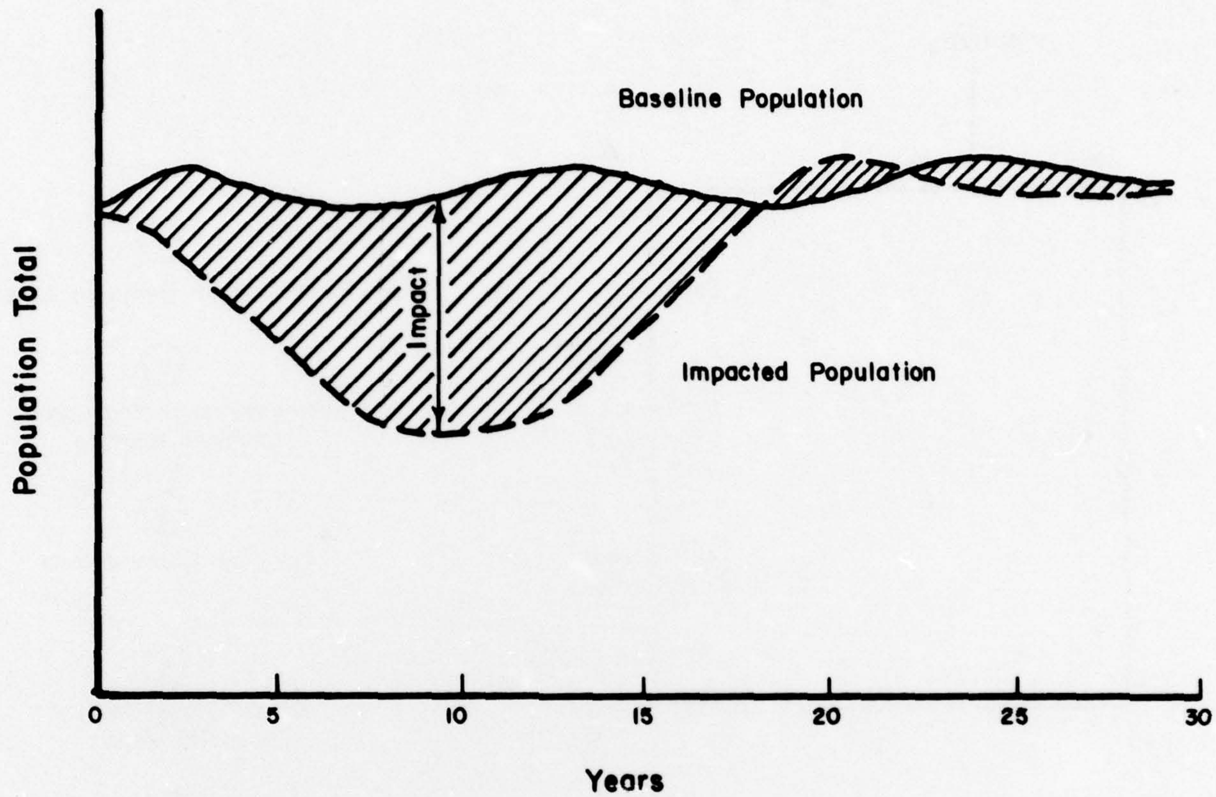
The proposed use of optimization techniques in RTV methodology is somewhat constrained by model complexities. RTVs are mathematically complex and nonlinear (as all good ecosystem modeling tends to be). Not only must the validity of individual RTVs be examined in more detail, but further research must be done on linearizing RTVs or applying nonlinear optimization techniques. Nevertheless, application of optimization to RTVs is a necessary part of future RTV research.

As part of the development of RTVs for aquatic ecosystems, a preliminary software package has been developed to demonstrate the utility of RTV methodology to impact analysis. The first step in the development of population-level RTVs is the development of an environmental setting. All basic data input in this step are derived from readily available sources and provide a list of modifying factors which will affect aquatic populations. For example, identification of water hardness will provide a basis for determining the toxicity of any heavy metals which might be present in discharges from the proposed activity. The effects of toxicants are expressed in changes in the population reproductive rates of selected fish species, as determined by changes in fecundity or survivability. The result is a prediction of population trends over a specified time and the use of that data to determine impact (Figure 10). If population predictions indicate a drop in population levels below a trigger level (90 percent of the mean values from the literature), a threshold is assumed and the action is identified as causing significant impact.

The RTV Framework

The following describes the integration of the RTV methodologies into the ETIS. (Figure 8 shows the ETIS structure.)

Figure 11 shows a hypothetical path of effect (A) within an aquatic ecosystem resulting from an activity which emits pollutants. The various analysis steps required to describe the cause/effect relationships and predict impact levels are shown. Physical impacts follow the more direct paths (B). Analysis Step 1 relates activities to emission levels or degree of physical impact. Step 2 provides for routing the pollutants across watersheds or through stream chan-



$$\text{Relative Impact} = \frac{\text{Impacted} - \text{Baseline}}{\text{Baseline}}$$

Figure 10. Sample population effects from population model.

nels. The resulting concentration of pollutants at some point in the stream system is determined in Step 3. This analysis step accounts for effects of dilution, chemical modification, or other processes which may increase or decrease the toxicity of the pollutants. Steps 4 and 5 provide analysis of direct and indirect effects.

Aquatic RTVs for pollutant emission are most effectively applied after Steps 1, 4, and 5. Water quality standards are the RTVs to be used after Step 1. A variety of RTVs could be used with the results in Steps 4 and 5. As noted previously, population levels are the most appropriate RTVs for initial development.

6 CONCLUSIONS

RTVs can be used as decision-making criteria for evaluating the significance of impacts on attributes of the aquatic ecosystem and for evaluating various alternatives to a new project or activity. The various constraints which impact RTV development can be classified as objective, operational, and model-related. Water quality standards and population levels are the most practical types of aquatic RTVs for initial development purposes. The approach presented here determines toxicity levels resulting from the introduction of pollutants into an aquatic ecosystem and expresses the effect of toxicants on population levels of various species. This information is used to

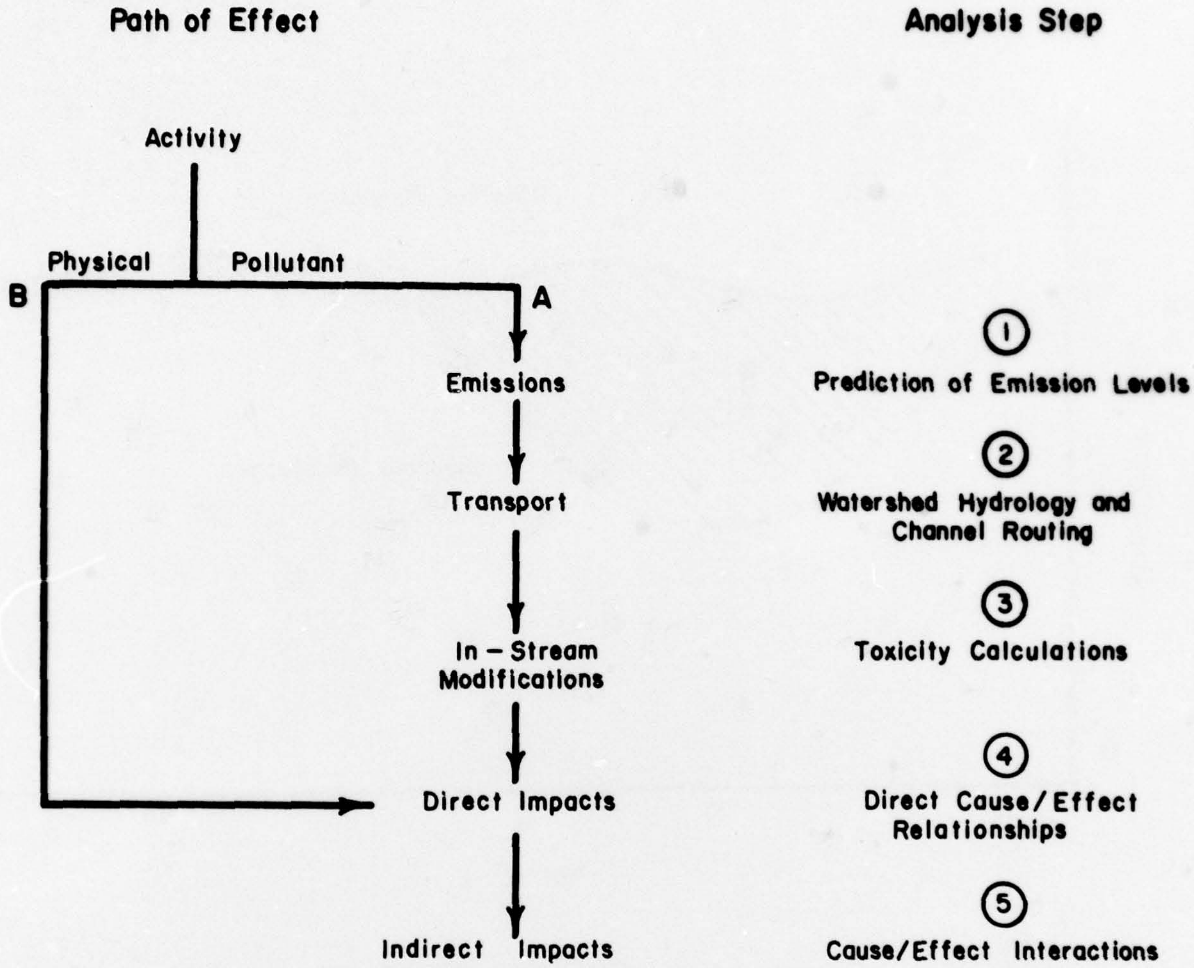


Figure 11. Path of effect and analysis steps.

establish the RTV for various pollutants and the significance of their input on aquatic features.

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